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Short running head title: Benefits of Attaining the 8-hour Ozone Standard

Key Words: air pollution, ozone, health impact assessment, standards, benefit analysis

Abbreviations:

AHRQ: Agency for Healthcare Research and Quality
AQS: Air Quality System
BenMAP: Environmental Benefits Mapping and Analysis Program
CAPMS: Criteria Air Pollutant Modeling System
CDC: Centers for Disease Control
CDC WONDER: Centers for Disease Control Wide-Ranging Online Data for Epidemiological Research
CI: Confidence interval
COI: Cost of illness
COPD: Chronic obstructive pulmonary disease
CPI-U: Consumer price index – urban
EPA: Environmental Protection Agency
ER: Emergency room
HIS: National Health Interview Survey
ICD: International Classification of Disease
MRAD: Minor restricted activity days
NCHS: National Center for Health Statistics
NHDS: National Hospital Discharge Survey
NHAMCS: National Hospital Ambulatory Medical Care Survey
NMMAPS: National Morbidity, Mortality and Air Pollution Study
NOx: Nitrogen oxides
PM_{10}: Particulate matter less than or equal to 10 microns
$PM_{2.5}$: Particulate matter less than or equal to 2.5 microns
ppb: parts per billion
POC: Parameter occurrence code
RIA: Regulatory impact analysis
SO_2: Sulfur dioxide
U.S.: United States
VOC: Volatile organic compounds
VNA: Voronoi neighbor averaging
VSL: Value of statistical life
WHO: World Health Organization
WTP: Willingness to pay

Outline:

I. Abstract:

During the 2000-2002 time period, between 36 and 56 percent of ozone monitors each year in the U.S. failed to meet the current ozone standard of 80 ppb for the 4^{th} highest maximum 8-hour ozone concentration. We estimated the health benefits of attaining the ozone standard at these monitors using the Environmental Protection Agency's Environmental Benefits Modeling and Analysis Program (BenMAP). We used health impact functions based on published epidemiological studies, and valuation functions derived from the economics literature. The estimated health benefits for 2000 and 2001 are similar in magnitude, while the results for 2002 are roughly twice that of each of the prior two years. The simple average of health impacts across the three years includes reductions of 800 premature deaths, 4,500 hospital and emergency room admissions, 900,000 school absences, and over a million minor restricted activity days. The simple average of benefits (including premature mortality) across the three years is $5.7 billion (90% CI: 0.6, 15.0) for the quadratic rollback simulation method and $4.9 billion (90% CI: 0.5, 14.0) for the proportional rollback simulation method. Results are sensitive to the form of the standard and to assumptions about background ozone levels. If the form of the standard is based on the 1^{st} highest maximum 8-hour concentration, impacts are increased by a factor of two to three. Increasing the assumed hourly background from zero to 40 ppb reduced impacts by 30 and 60 percent for the proportional and quadratic attainment simulation methods, respectively.

II. Introduction

The Clean Air Act(U.S. EPA 1970) identifies tropospheric ozone as one of six "criteria pollutants" – pervasive pollutants considered harmful to human health. Tropospheric ozone forms as a result of atmospheric reactions of nitrogen oxides (NOx) and volatile organic compounds (VOCs) in the presence of sunlight. Both local emissions sources, such as traffic, and emissions transported from upwind sources, such as electric utilities, contribute to ambient ozone levels in populated areas.

In 1997, EPA changed the ozone standard to 80 ppb to reflect new scientific studies showing that ozone causes health effects at levels lower than the previous 120 ppb standard. Additionally, the form of the standard was changed to reflect studies showing that exposure times longer than one hour are of concern. EPA set the form of the standard, which is the threshold for compliance and violations, at the 4th highest daily maximum 8-hour average occurring each year, averaged over a three-year period.

New scientific studies published since 1996 have increased the body of evidence supporting the association between ambient ozone and a number of serious health effects (WHO 2004). For example, studies examining the association between ambient ozone and premature mortality have increased the weight of evidence supporting this important health impact (Ito and Thurston 2001; Anderson et al. 2004).

Our purpose for this analysis is to assess the human health benefits of attaining the 8-hour ozone standard. We apply a damage function approach similar to those used in several recent U.S. EPA regulatory impact analyses, including those for the proposed Clean Air Interstate Rule and the final Clean Air Nonroad Diesel Rule (U.S. EPA 2004a, 2004b). We focused the assessment on the benefits that might have been achieved if current monitored ozone levels (represented by the years 2000-2002) were reduced just to the levels required to meet the 8-hour standard. We conducted analyses to examine the sensitivity of our results to a number of different assumptions about the form of the standard, background levels of ozone, methods for simulating attainment of the 8-hour ozone standard, and the choice of health effects and effect estimates from published epidemiological studies.

The remainder of this paper provides detailed descriptions of the data and methods in this analysis, along with the results. Section III describes monitored ozone levels in 2000, 2001 and 2002, provides details on how we assigned monitored ozone levels to populations to estimate population-level exposures, and outlines the two approaches we used to simulate attainment. Section IV discusses the literature on ozone-related health effects and describes the specific set of health impact functions we used in the benefits analysis. Section V describes the economic values selected to estimate the dollar value of ozone-related health impacts. Section VI discusses how we address uncertainty in the analysis. Finally, Section VII presents the results and implications of the analysis.

III. Simulation of Changes in Population-Level Exposures to Ambient Ozone Due to Attainment

a. Selecting Monitoring Data

To estimate population-level ozone concentrations, we began by obtaining ozone monitoring data from EPA's Air Quality System (AQS), a database of ambient air pollution data collected by EPA, state, local and tribal air pollution control agencies from over a thousand monitoring stations across the country. We analyzed these data using the environmental Benefits Mapping and Analysis Program (BenMAP), a program developed by the U.S. Environmental Protection Agency for use in estimating the health impacts and economic benefits associated with changes in ambient air pollution (you can obtain a copy of BenMAP by e-mailing a request to hubbell.bryan@epa.gov). We used SAS (release 8.02, Cary, NC) to process the AQS data for use in BenMAP.

In order to characterize ozone levels, we selected monitors following criteria generally consistent with those EPA uses to determine attainment and nonattainment of the 8-hour standard. We selected monitors that had a sufficient number of observations during the ozone "season," which stretches from May 1 through September 30. There are 153 days in this period. Many areas of the United States, including Southern California and Texas, have a longer ozone season. Accounting for the longer ozone season in these areas would lead to an increase in the estimated benefits of attaining the standards.

Because missing monitor observations are common, we selected only those monitors that had observations on at least half the days in this period. Specifically, each monitor had to have at least 77 valid days, with a valid day defined as having at least nine hourly observations between 8 a.m. and 7:50 p.m. We did not use data from any monitor with a parameter occurrence code (POC) greater than four to avoid errors that may be introduced by using non-standard monitors. (POC codes are used to distinguish among multiple monitors at the same site that are measuring the same parameter. In general, a higher POC code is assigned to monitors that are not the primary ozone monitor). For those locations with more than one ozone monitor, we selected the monitor with the lowest POC code (e.g., we chose POC 1 rather than 2), and dropped any others.

Table 1 summarizes the distribution of monitored 4th highest maximum daily 8-hour average ozone concentrations across the three study years. In all years, at least 35 percent of monitors failed to meet the level of the standard. However, there was some variability between years in the proportion of non-attainment monitors and in the amount by which monitors exceeded the standard. In 2000 and 2001, less than 40 percent of monitors exceeded the standard, and 5 percent or less of the monitors exceeded 100 ppb. In 2002, 56 percent of monitors had ozone levels that exceeded the standard, and 14 percent had ozone levels above 100 ppb. Monitored ozone levels in 2002 were higher in part due to meteorological conditions favorable for ozone formation and transport of ozone precursors (U.S. EPA 2003).

Ozone concentrations show spatial patterns, with certain areas of the United States, including California, having consistently high ozone values from 2000 through 2002. Other areas, such as the Southeast and Northeast, varied a great deal across those years. This may result from: differences in climatic variability; natural phenomena, such as wildfires; or differences in ozone precursor (NOx and VOC) emissions. Year-to-year precursor emissions may vary due to economic cycles; changes in electricity generation, such as switching from coal to natural gas; or changes in vehicle usage.

b. Applying Spatial Interpolation

Monitor data represent ambient ozone levels at a series of discrete points in space. However, benefits analysis requires an estimate of ambient ozone concentrations for populations across the United States. For each year of monitoring data (2000, 2001 and 2002), we generated estimates of average ambient ozone levels for every county in the U.S using applied spatial interpolation methods. Our base case analysis uses Voronoi neighbor averaging (VNA), an algorithm that estimates ambient ozone levels by selecting the closest neighboring monitors surrounding the center of each county and then calculating the inverse distance weighted average of the monitor values for the selected neighboring monitors (see for example Chen, Zhao and Li 2004; Gold 1997). This method provides a relatively smooth surface in densely monitored areas.

We analyzed the accuracy of the VNA interpolation procedure by dropping individual monitors and predicting their ambient ozone levels using the remaining monitors. The national average differences between predicted and observed annual averages less than one percent in all cases, with standard deviations ranging from ten to twelve percent. The largest differences occurred in rural areas and large portions of the western United States where few monitors are present – ozone estimates in these cases are often based partially on monitors that are quite distant.

Most populations live within fifty kilometers of an ozone monitor, however, so we can be reasonably confident that estimates of ambient ozone levels will be acceptable for most populated areas. We explored the sensitivity of the results to the choice of spatial interpolation method by estimating ambient ozone levels using a distance limited version of VNA (where all monitors further than fifty kilometers are discarded when choosing neighbors), as well as using a simple closest monitor assignment. A detailed explanation of each of these methods is provided in the Supplemental Material.

c. Reducing ozone levels to meet the standard

In order to demonstrate the benefits of attaining the 8-hour standard in 2000, 2001 and 2002, we specified how ozone levels would be reduced to bring the specific attainment "metric" (4^{th} highest daily maximum 8-hour average) down to the level of the standard. EPA's primary (for health protection) and secondary (for environmental and welfare protection) 8-hour ozone standards both are 80 ppb. In determining attainment and nonattainment, however, the Agency must use rounding. As a result, we consider ozone values up to, and including, 84 ppb as meeting the standard.

There is more than one way to reduce the distribution of hourly ozone values to simulate attainment. (It should be noted that for simplicity we treated the form of the standard as simply the 4^{th} highest daily maximum 8-hour average, rather than the 4^{th} highest daily maximum 8-hour average *averaged over the three previous years*.)

We investigated two different methods: percentage (or proportional) rollback; and quadratic rollback. Percentage rollback simply reduces all daily metric values by the percentage required to bring the violating day (the day with the 4^{th} highest value) down to 84 ppb. The quadratic rollback method reduces larger metric values proportionally more than smaller values.

It is not clear which method provides a more realistic simulation of an attainment strategy. If control strategies affect emissions on all days during the ozone season, then using percentage rollback may be appropriate. If control strategies affect emissions on days with higher ozone levels more than days with lower levels, then quadratic rollback may be more realistic. Both of these approaches represent implementation strategies areas may select to meet the ozone standard. See Supplemental Material for more details on the two methods.

For both methods, we assume a constant background 8-hour daily maximum ozone level of 40 ppb, representing the amount of ozone (for this averaging period) that is not attributable to U.S. anthropogenic sources. It is assumed that this background cannot be affected by attempts to attain the ozone standards, and thus this portion of the estimated ambient ozone levels is not adjusted by either rollback method.

Vingarzan (2004) surveyed recent literature on background ozone concentrations and concluded that based on data from 1983 to 2001, median background levels in the United States ranged between 13 and 47 ppb. Vingarzan (2004) notes that background levels appear to be increasing over time due to increased contributions from international transport of ozone precursors. As such, we selected a background ozone level towards the upper end of the observed range, since our monitor data is based on later years. The background level likely varies across the U.S., and our assumption of 40 ppb adds uncertainty to the analysis (Vingarzan 2004). We investigate the impact of different assumptions about background levels of the attainment metric in a sensitivity analysis.

Once BenMAP has calculated how the attainment metric will be affected for each day, it calculates how the other ozone metrics required for the various health impact functions will be affected. These include daily maxima for one and eight hour periods, as well as daily averages over different time periods, including the 24 hour average, the 5 hour average (10:00 AM – 2:50 PM), and the 8 hour average (9:00 AM – 4:50 PM). To do this, BenMAP rolls back individual hourly ozone observations such that they meet the target metric values. For details on this process, see Supplemental Material.

In adjusting individual hourly ozone values to meet the target metric value, we assumed that there is no fixed background level of ozone for any particular hour and set the background to zero. Any given hourly value may have a specific background component; however, we are unable to determine what this component might be. We examined the impact of assuming alternative hourly background levels as a sensitivity analysis.

Finally, BenMAP uses the adjusted hourly values to calculate the adjusted ozone summary measures, e.g. 24-hour average, one-hour maximum, etc. Using the three methods described earlier, BenMAP then spatially interpolates the set of adjusted summary measures to the center of each county. The differences between the spatially interpolated baseline and the adjusted summary measures are the basic air quality inputs to the health benefits model.

Note that BenMAP does not adjust monitors that meet the attainment test (those with 4^{th} highest maximum daily 8-hour average at 84 ppb or below). However, these monitors are included in the interpolation process, so that the ozone levels assigned to a population in a given county will, in most cases, reflect an average of monitors with ozone reductions and those with no reduction. In reality, there will be reductions in ozone levels at monitors in a nonattainment area due to controls applied to meet the standard. As such, we are likely underestimating the change in ambient ozone that would occur as the result of implementing attainment strategies.

IV. Health Impact Functions

Health impact functions measure the change in a health endpoint of interest, such as hospital admissions, for a given change in ozone. Health impact functions are derived from the epidemiology literature. A standard health impact function has four components: 1) an effect estimate from a particular epidemiological study; 2) a baseline incidence rate for the health effect (obtained from either the epidemiology study or a source of public health statistics such as the Centers for Disease Control); 3) the affected population; and 4) the estimated change in the relevant ozone summary measures.

A typical health impact function might look like:

$$\Delta y = y_0 \cdot (e^{\beta \cdot \Delta x} - 1),$$

where y_0 is the baseline incidence, equal to the baseline incidence rate times the potentially affected population, β is the effect estimate, and Δx is the estimated change in the summary ozone measure. There are other functional forms, but the basic elements remain the same. Section III described the ozone air quality inputs to the health impact functions. The following subsections describe the sources for each of the other elements: affected populations; effect estimates; and baseline incidence rates.

a. Affected Populations

The starting point for estimating affected populations is the 2000 U.S. Census block level dataset (Geolytics 2002). BenMAP incorporates 250 age/gender/race categories to match specific populations potentially affected by ozone and other air pollutants. The software constructs specific populations matching the populations in each epidemiological study by accessing the appropriate age-specific populations from the overall population database. BenMAP projects populations to 2001 and 2002 using growth factors based on economic projections (Woods and Poole Inc. 2001).

b. Effect Estimate Sources

The most significant benefits of reducing ambient concentrations of ozone are attributable to reductions in health risks. EPA's Ozone Criteria Document and the World Health Organization's 2003 and 2004 reports outline numerous health effects known or suspected to be linked to exposure to ambient ozone (US EPA 1996b; WHO 2003; Anderson et al. 2004).

More than one thousand new health and welfare studies have been published since EPA issued the 8-hour ozone standard in 1997. Many of these studies investigated the impact of ozone exposure on health effects such as: changes in lung structure and biochemistry; lung inflammation; asthma exacerbation and causation; respiratory illness-related school absence; hospital and emergency room visits for asthma and other respiratory causes; and premature death.

We excluded some health effects from this analysis for four reasons: (1) the possibility of double counting (such as hospital admissions for specific respiratory diseases); (2) uncertainties in applying effect relationships that are based on clinical studies to the affected population; (3) a lack of an established concentration-response relationship; or 4) the inability to appropriately value the effect (for example, changes in forced expiratory volume) in economic terms. Table 2 lists the health endpoints included in the primary and sensitivity analyses for this paper.

In selecting epidemiological studies as sources of effect estimates, we applied several criteria to develop a set of studies that is likely to provide the best estimates of impacts in the U.S. To account for the potential impacts of different health care systems or underlying health status of populations, we give preference to U.S. studies over non-U.S. studies. In addition, due to the potential for confounding by co-pollutants, we give preference to effect estimates from models including both ozone and particulate matter over single pollutant models.

A number of endpoints that are not health-related also may significantly contribute to monetized benefits. These include: decreased outdoor worker productivity; decreased yields for commercial and non-commercial crops; decreased commercial forest productivity; damage to urban ornamental plants; impacts on recreational demand from damaged forest aesthetics; and damage to ecosystem functions (U.S. EPA 1996a, 1999). Estimation of these impacts is beyond the scope of this analysis.

Effect Estimates: Premature Mortality

While particulate matter is the air pollutant most clearly associated with premature mortality, recent research suggests that repeated ozone exposure likely contributes to premature death. Several recent analyses have found consistent statistical associations between ozone exposure and increased mortality (Fairly et al. 2003; Thurston and Ito 1999; Toulomi et al. 1997). In addition, while the 2000 National Morbidity, Mortality and Air Pollution Study, known as NMMAPS, did not find an effect of ozone on total mortality across the full year, Dominici et al. (2000), who conducted the study, did observe an effect after limiting the analysis to summer, when ozone levels are highest.

Although they do not constitute a database as extensive as that for particulate matter, these recent studies provide supporting evidence for including mortality in ozone health benefits analyses. In a 2001 analysis, Thurston and Ito reviewed previously published time-series studies examining the effect of daily ozone levels on daily mortality. Thurston and Ito hypothesized that much of the variability in published estimates of the ozone/mortality effect could be explained by how well each model controlled for the influence of weather, an important confounder; and that earlier studies, which used less-sophisticated approaches to controlling for weather, consistently under-predicted the ozone/mortality effect.

Thurston and Ito (2001) also found that models incorporating a non-linear temperature specification appropriate for the "U-shaped" nature of the temperature/mortality relationship (i.e., increased deaths at both very low and very high temperatures) produced ozone/mortality effect estimates that were both more strongly positive (a 2 percent increase in relative risk over the pooled estimate for all studies evaluated) and consistently statistically significant. Further accounting for the interaction effects between temperature and relative humidity strengthened the positive effect. Including a particulate matter index to control for PM/mortality effects had little effect on these results, suggesting a relationship between ozone and mortality independent of that for PM. However, most of the studies Thurston and Ito examined controlled only for PM_{10} or broader measures of particles and did not directly control for $PM_{2.5}$. As such, there still may be potential for confounding of $PM_{2.5}$ and ozone mortality effects, given that ozone and $PM_{2.5}$ are highly correlated during summer months in some areas.

Two recent World Health Organization reports found that "recent epidemiological studies have strengthened the evidence that there are short-term O_3 effects on mortality and respiratory morbidity and provided further information on exposure-response relationships and effect modification." (WHO 2003, 2004) In addition, Levy et al. (2001) assessed the epidemiological evidence regarding the link between short-term exposures to ozone and premature mortality. Based on four U.S. studies (Kellsall et al. 1997; Moolgavkar et al. 1995a; Ito and Thurston 1996; Moolgavkar 2000), they concluded that an appropriate pooled effect estimate is a 0.5 percent increase in premature deaths per 10 $\mu g/m^3$ increase in 24-hour average ozone concentrations, with a 95 percent confidence interval between 0.3 percent and 0.7 percent.

We included ozone mortality in the base health effects estimate for the ozone benefits reanalysis, with the recognition that the exact magnitude of the effects estimate is subject to continuing uncertainty. We used results from three U.S. studies to calculate the base-case ozone mortality estimate. We selected these studies (Ito and Thurston 1996; Moolgavkar et al. 1995a; Samet et al. 1997) based on the logic that the demographic and environmental conditions existing when these studies were conducted would, on average, be most similar (relative to international studies) to the conditions prevailing when the ozone standards would be implemented. We examined the impact of including a fourth U.S. study by Kinney et al. (1995) in a sensitivity analysis. We excluded Kinney et al. from the primary analysis, because, as Levy et al. (2001) noted, that study included only a linear term for temperature. Because Kinney et al. (1995) found no significant ozone effect, including this study in the primary analysis would lead to an underestimate of true mortality impacts and increase the uncertainty surrounding the estimated mortality reductions.

We then estimated the change in mortality incidence resulting from application of the effect estimate from each study and combined the results using a random-effects weighting procedure, discussed in Supplemental Material, that accounts for both the precision of the individual effect estimates as well as between study variability. However, it is important to note that this procedure only captures the uncertainty in the underlying epidemiological work, and does not capture other sources of uncertainty, such as that in the estimation of air pollution exposure (Levy et al. 2000).

Effect Estimates: Respiratory Hospital Admissions

Detailed hospital admission and discharge records provide data for an extensive body of literature examining the relationship between hospital admissions and air pollution. This is especially true for the population 65 and older, because of the availability of detailed Medicare records. Because the number of hospital admission studies is so large, we used results from a number of studies to pool some hospital admission endpoints. In addition, there is one study (Burnett et al. 2001) providing an effect estimate for respiratory hospital admissions in children under two.

To estimate total respiratory hospital admissions associated with changes in ozone for adults over 65, we first estimated the change in hospital admissions for the separate effects categories each study provided for each city, including Minneapolis, Detroit, Tacoma and New Haven. To estimate all-respiratory hospital admissions for Detroit, we added the pneumonia and COPD estimates, based on the effect estimates in the Schwartz study (1994b). Similarly, we summed the estimated hospital admissions based on the effect estimates the Moolgavkar study reported for Minneapolis (Moolgavkar et al. 1997). To estimate all-respiratory hospital admissions for Minneapolis using the Schwartz study (1994a), we simply estimated pneumonia hospital admissions based on the effect estimate. Making this assumption that pneumonia admissions represent the total impact of ozone on hospital admissions will give some weight to the possibility that there is no relationship between ozone and COPD, reflecting the equivocal evidence represented by the different studies. We then used a fixed effects pooling procedure to combine the two all-respiratory hospital admission estimates for Minneapolis. Finally, we used random effects pooling to combine the results for Minneapolis, Detroit, in addition to results from studies in Tacoma and New Haven. As noted above, this pooling approach accounts for both the precision of the individual effect estimates as well as between study variability characterizing differences across study locations.

Effect Estimate: Asthma-Related Emergency Room Visits

We used three studies as the source for the concentration-response functions we used to estimate the effects of ozone exposure on asthma-related emergency room (ER) visits: Cody et al. (1992); Weisel et al. (1995); and Stieb et al. (1996). We estimated the change in ER visits using the effect estimate from each study and then pooled the results using the random effects pooling procedure described in Supplemental Material. A more recent study by Jaffe et al. (2003) examined the relationship between ER visits and air pollution for people ages five to 34 in the Ohio cities of Cleveland, Columbus and Cincinnati from 1991 through 1996. We did not use this particular study in our primary analysis, because it represents a more limited population and excludes potentially important impacts on the over-35 population. However, because many asthma-related ER visits involve children, this study is included in a sensitivity analysis showing the magnitude of results for all ages relative to those for a population more heavily weighted toward children. We include both hospital admissions and ER visits as separate endpoints associated with ozone exposure, because our estimates of hospital admission costs do not include the costs of ER visits.

Effect Estimate Sources: Minor Restricted Activity Days

Minor restricted activity days (MRADs) occur when individuals reduce most usual daily activities and replace them with less-strenuous activities or rest, but do not miss work or school. We estimated the effect of ozone on MRADs using a concentration-response function derived from Ostro and Rothschild (1989). These researchers estimated the impact of ozone and $PM_{2.5}$ on MRAD incidence in a national sample of the adult working population (ages 18 to 65) living in metropolitan areas. We developed separate coefficients for each year of the Ostro and Rothschild analysis (1976-1981), which we then combined for use in EPA's analysis. The effect estimate used in the impact function is a weighted average of the coefficients in Ostro and Rothschild (1989, Table 4), using the inverse of the variance as the weight.

Effect Estimate: School Absences

Children may be absent from school due to respiratory or other acute diseases caused, or aggravated by, exposure to air pollution. Several studies have found a significant association between ozone levels and school absence rates. We use two recent studies (Gilliland et al. 2001; Chen et al. 2000) to estimate changes in school absences resulting from changes in ozone levels. The Gilliland et al. study estimated the incidence of new periods of absence, while the Chen et al. study examined absence on a given day. We converted the Gilliland et al. estimate to days of absence by multiplying the absence periods by the average duration of an absence. We estimated 1.6 days as the average duration of a school absence, the result of dividing the average daily school absence rate from Chen et al. (2000) and Ransom and Pope (1992) by the episodic absence rate from Gilliland et al. Thus, each Gilliland et al. period of absence is converted into 1.6 absence days.

Following recent advice from the National Research Council (2002), we calculated reductions in school absences for the full population of school age children, ages five to 17. This is consistent with recent peer-reviewed literature on estimating the impact of ozone on school absences (Hall et al. 2003). We estimated the change in school absences using both Chen et al. (2000) and Gilliland et al. (2001) and then pooled the results using the random effects pooling procedure described in Supplemental Material.

c. Baseline Incidence Rates

Epidemiological studies of the association between pollution levels and adverse health effects generally provide a direct estimate of the relationship of air quality changes to the *relative risk* of a health effect, rather than estimating the absolute number of avoided cases. For example, a typical result might be that a 100 ppb decrease in daily ozone levels might, in turn, decrease hospital admissions by 3 percent. The baseline incidence of the health effect is necessary to convert this relative change into a number of cases. A baseline incidence rate is the estimate of the number of cases of the health effect per year in the assessment location, as it corresponds to baseline pollutant levels in that location. To derive the total baseline incidence per year, this rate must be multiplied by the corresponding population number. For example, if the baseline incidence rate is the number of cases per year per 100,000 people, that number must be multiplied by the number of 100,000s in the population.

Table 3 summarizes the sources of baseline incidence rates and provides average incidence rates for the endpoints included in the analysis. For both baseline incidence and prevalence data, we used age-specific rates where available. We applied concentration-response functions to individual age groups and then summed over the relevant age range to provide an estimate of total population benefits. In most cases, we used a single national incidence rate, due to a lack of more spatially disaggregated data. Whenever possible the rates used are national averages, because these data are most applicable to a national assessment of benefits. For some studies, however, the only available incidence information comes from the studies themselves; in these cases, incidence in the study population is assumed to represent typical incidence at the national level. Regional incidence rates are available for hospital admissions, and county-level data are available for premature mortality.

V. Economic Values for Health Outcomes

Reductions in ambient concentrations of air pollution generally lower the risk of future adverse health effects for a large population. Therefore, the appropriate economic measure is willingness-to-pay (WTP) for changes in risk of a health effect rather than WTP for a health effect that would occur with certainty (Freeman 1993). Epidemiological studies generally provide estimates of the relative risks of a particular health effect that is avoided because of a reduction in air pollution. We converted those to units of avoided statistical incidence for ease of presentation. We calculated the value of avoided statistical incidences by dividing individual WTP for a risk reduction by the related observed change in risk. For example: suppose a pollution-reduction regulation is able to reduce the risk of premature mortality from 2 in 10,000 to 1 in 10,000 (a reduction of 1 in 10,000). If individual WTP for this risk reduction is \$100, then the WTP for an avoided statistical premature death is \$1 million (\$100/0.0001 change in risk).

WTP estimates generally are not available for some health effects, such as hospital admissions. In these cases, we used the cost of treating or mitigating the effect as a primary estimate. These cost-of-illness (COI) estimates generally understate the true value of reducing the risk of a health effect, because they reflect the direct expenditures related to treatment, but not the value of avoided pain and suffering (Harrington and Portney 1987; Berger 1987). We provide unit values for health endpoints (along with information on the distribution of the unit value) in Table 4. All values are in constant year 2000 dollars, adjusted for growth in real income. Economic theory argues that WTP for most goods (such as environmental protection) will increase if real income increases. Many of the valuation studies used in this analysis were conducted in the late 1980s and early 1990s. Because real income has grown since the studies were conducted, people's willingness to pay for reductions in the risk of premature death and disease likely has grown as well. We did not adjust cost of illness-based values, because they are based on current costs. Similarly, we did not adjust the value of school absences, because that value is based on current wage rates. Table 4 presents the values for individual endpoints adjusted to year 2000 income levels.

Mortality

To estimate the monetary benefit of reducing the risk of premature death, we used the "value of statistical lives" saved (VSL) approach, which is a summary measure for the value of small changes in mortality risk for a large number of people. The VSL approach applies information from several published value-of-life studies to determine a reasonable monetary value of preventing premature mortality. The mean value of avoiding one statistical death is estimated to be roughly $6 million in year 2000 dollars (2000 $). This represents an intermediate value from a variety of estimates in the economics literature (see U.S. EPA 1999).

Respiratory Hospital Admissions

In the absence of estimates of societal WTP to avoid hospital visits/admissions for specific illnesses, estimates of total cost of illness (total medical costs plus the value of lost productivity) typically are used as conservative, or lower bound, estimates. These estimates are biased downward, because they do not include the willingness-to-pay value of avoiding pain and suffering.

The International Classification of Diseases (ICD-9, 1979) code-specific COI estimates we used in this analysis consist of estimated hospital charges and the estimated opportunity cost of time spent in the hospital, (based on the average length of a hospital stay for the illness). We based all estimates of hospital charges and length of stays on statistics provided by the Agency for Healthcare Research and Quality (AHRQ 2000). We estimated the opportunity cost of a day spent in the hospital as the value of the lost daily wage, regardless of whether the hospitalized individual is in the workforce. To estimate the lost daily wage, we divided the 1990 median weekly wage by five and inflated the result to year 2000$ using the CPI-U "all items." The resulting estimate is $109.35. The total cost-of-illness estimate for an ICD code-specific hospital stay lasting *n* days, then, was the mean hospital charge plus $109 • *n*.

Asthma-Related Emergency Room Visits

To value asthma emergency room visits, we used a simple average of two estimates from the health economics literature. The first estimate comes from Smith et al. (1997), who reported approximately 1.2 million asthma-related emergency room visits in 1987, at a total cost of $186.5 million (1987$). The average cost per visit that year was $155; in 2000$, that cost was $311.55 (using the CPI-U for medical care to adjust to 2000$). The second estimate comes from Stanford et al. (1999), who reported the cost of an average asthma-related emergency room visit at $260.67, based on 1996-1997 data. A simple average of the two estimates yields a (rounded) unit value of $286.

Minor Restricted Activity Days

No studies are reported to have estimated WTP to avoid a minor restricted activity day. However IEc (1993) has derived an estimate of willingness to pay to avoid a minor *respiratory* restricted activity day, using estimates from Tolley et al. (1986) of WTP for avoiding a combination of coughing, throat congestion and sinusitis. The IEc estimate of WTP to avoid a minor respiratory restricted activity day is $38.37 (1990$), or about $52 ($2000).

Although Ostro and Rothschild (1989) statistically linked ozone and minor restricted activity days, it is likely that most MRADs associated with ozone exposure are, in fact, minor *respiratory* restricted activity days. For the purpose of valuing this health endpoint, we used the estimate of mean WTP to avoid a minor respiratory restricted activity day.

School Absences

To value a school absence, we: (1) estimated the probability that if a school child stays home from school, a parent will have to stay home from work to care for the child; and (2) valued the lost productivity at the parent's wage. To do this, we estimated the number of families with school-age children in which both parents work, and we valued a school-loss day as the probability that such a day also would result in a work-loss day. We calculated this value by multiplying the proportion of households with school-age children by a measure of lost wages.

We used this method in the absence of a preferable WTP method. However, this approach is likely to understate the value of school-loss days in three ways: First, it omits willingness to pay to avoid the symptoms/illness that resulted in the school absence; second, it effectively gives zero value to school absences that do not result in work-loss days; and third, it uses conservative assumptions about the wages of the parent staying home with the child.

For this valuation approach, we assumed that in a household with two working parents, the female parent will stay home with a sick child. From the Statistical Abstract of the United States (U.S. Census Bureau, 2001), we obtained: (1) the numbers of single, married and "other" (widowed, divorced or separated) working women with children; and (2) the rates of participation in the workforce of single, married and "other" women with children. From these two sets of statistics, we calculated a weighted average participation rate of 72.85 percent.

Our estimate of daily lost wage (wages lost if a mother must stay at home with a sick child) is based on the year 2000 median weekly wage among women ages 25 and older (U.S. Census Bureau, 2001). This median weekly wage is $551. Dividing by five gives an estimated median daily wage of $103. To estimate the expected lost wages on a day when a mother has to stay home with a school-age child, we first estimated the probability that the mother is in the workforce then multiplied that estimate by the daily wage she would lose by missing a work day: 72.85 percent times $103, for a total loss of $75.

VI. Methods for Describing Uncertainty

Any complex analysis is likely to reflect many sources of uncertainty, and this analysis is no exception. We used numerous inputs to derive the benefits estimate, including: measured ozone concentrations at monitor sites; interpolation methods; estimates of values (both from willingness-to-pay and cost-of-illness studies); population estimates; baseline incidence rate estimates; and income estimates. Each of these inputs may be uncertain, and depending on its location in the benefits analysis, each may have a disproportionately large impact on final estimates of total benefits. For example, we used measured ozone concentrations at monitor sites in the first stage of the analysis, meaning that any uncertainty in those measurements will propagate as the analysis continues. When compounded with uncertainty in later stages of analysis, even small uncertainties in monitored ozone levels can lead to large impacts on total benefits.

Given the wide variety of sources for uncertainty and the potentially large degree of uncertainty about any specific estimate, we characterized uncertainty in two ways: through the use of a limited scope Monte Carlo analysis, and through sensitivity analyses.

More than one source of uncertainty usually exists, even for individual endpoints. This makes it difficult to provide an overall quantified uncertainty estimate, either for individual endpoints or total benefits. An example: the health impact function used to estimate avoided premature deaths has an associated standard error that represents the statistical error around the effect estimate in the underlying epidemiological study. In our results, we report a confidence interval based on this standard error, reflecting the uncertainty in the estimated change in incidence of avoided premature deaths. However, this confidence interval omits the contribution of air quality changes, baseline incidence rates, populations exposed and transferability of the effect estimate to diverse locations. As a result, the reported confidence interval gives a potentially misleading picture about the overall uncertainty in the estimates. This information should be interpreted within the context of the larger uncertainty surrounding the entire analysis.

We used Monte Carlo methods to generate confidence intervals around the estimated health impact and dollar benefits. Monte Carlo simulation uses random sampling from distributions of parameters to characterize the effects of uncertainty on output variables, such as incidence of premature mortality. Distributions for individual effect estimates are based on the reported standard errors in the epidemiological studies. Distributions for unit values are described in Table 4.

VII. Results and Implications

Table 5 summarizes the incidence and valuation for each year associated with two attainment simulation methods, percentage and quadratic. Table 6 provides the results averaged across the three years. In addition to the mean incidence and valuation estimates, we have included a 5^{th} and 95^{th} percentile estimate in Table 6, based on the Monte Carlo simulations described above. To calculate the air quality values under each attainment scenario, we rolled back the ozone monitor data so that the 4^{th} highest daily maximum 8-hour average just met the level required to attain the standard. This approach will likely understate the benefits that would occur due to implementation of actual controls to reduce ozone precursor emissions. These controls would likely result in reductions in ozone concentrations at all monitors within a non-attainment area, rather than just at those monitors with out of attainment ozone values. As such, our results are an underestimate of the likely benefits of attaining the ozone standard. In all of the primary analytical cases, we used Voronoi Neighbor Averaging with no distance limit, and assumed a 40 ppb background level for the attainment metric and an hourly background level of zero.

The results for 2000 and 2001 are similar in magnitude, while the results for 2002 are roughly twice that of each of the prior two years. The simple average of benefits (including premature mortality) across the three years is $5.7 billion (90% CI: 0.6, 15.0) for the quadratic rollback simulation method and $4.9 billion (90% CI: 0.5, 14.0) for the percentage rollback simulation method. Average benefits without premature mortality are $200 million (90% CI: 72, 350) for the quadratic rollback method and $160 million (90% CI: 65, 310) for the percentage rollback method. Including premature mortality in our estimates had the largest impact on the overall magnitude of benefits: Premature mortality benefits account for more than 95 percent of the total benefits we can monetize.

Table 7 shows the impact on incidence of health impacts of a range of assumptions regarding how we rolled back the ozone monitor values. We considered the impact of ordinality – that is of choosing the 1st versus the 4th highest daily maximum 8-hour average – and we chose a range of alternative background levels. Regardless of attainment simulation method, ordinality had the largest apparent impact, with roughly a factor of two to three separating results between the 1st highest and 4th highest 8-hour maximum. It is important to note that health impacts are likely to occur whenever the 8-hour daily maximum is elevated, not just when the number of exceedances is greater than four. While the standard reflects the underlying health science, and seeks to protect public health, it does not guarantee zero health impacts. Having said that, the magnitude of the difference in this analysis is still somewhat surprising.

There are two elements that contribute to this result. First, certain monitors will meet the standard with an ordinality of 4, but will not meet the standard with an ordinality of 1. That is, some monitors may have one metric value that exceeds 84 ppb, but will not have four such values. As discussed above, monitors that meet the standard are not adjusted at all, so these monitors will have a large impact on the results. Secondly, certain monitors have a small number of outlier metric values that are much higher than all of the rest. Because the rollback strategies both adjust all metric values, basing a rollback on these outlier values can cause much higher reductions across the entire year.

The impact of attainment metric background and the hourly background depended on attainment simulation method. Under the percentage rollback attainment simulation method, shifting the attainment metric background from 40 to 0 increased impacts by roughly a factor of two, but the same shift under the quadratic rollback method had no significant impact on results. However, shifting the *hourly* background level from 0 to 40 under the quadratic rollback method resulted in a roughly 60 percent reduction in impacts, while making the same background shift using the percentage rollback method reduced impacts by around a third.

For any particular assumption of background ozone levels, our estimates are likely to understate the actual benefits that would occur from implementing control strategies to attain the 8-hour standard. This is due to our assumption that only the specific monitors that are out of attainment in any area will realize reductions in ozone levels. Our estimates of benefits in areas of the country with longer ozone seasons, such as California and Texas, will also be underestimates due to our assumption of a fixed ozone season from May 1 to September 30 for the entire nation. Analyses of specific attainment strategies should allow for changes in ambient ozone across all monitors in a nonattainment area, as well as accounting for the variable length of the ozone season. Because there is currently no known threshold for most ozone-related health effects, there is likely to be a significant benefit to reducing ozone concentrations beyond the standard at monitors that currently attain the standard.

Applying a distance limit of 50 kilometers to the VNA method reduced benefits by 3 to 10 percent, depending on the year of analysis. Use of a closest monitor algorithm with a 50 kilometer limit reduced benefits by 10 to 15 percent, depending on the year of analysis. Most of this difference occurs because about 10 percent of the population lives more than 50 kilometers away from an ozone monitor. Detailed sensitivity analyses examining the choice of interpolation method are available on request.

Our estimates of mortality related benefits of attaining the standards may change, based on emerging meta-analyses of the ozone mortality literature. If these meta-analyses confirm the results of Thurston and Ito (2001), Levy et al (2001) or the WHO report (2003) meta-analyses, the mean mortality benefits may increase by a factor of 2, suggesting that reductions in premature mortality associated with attainment of the ozone standards might be as high as 1,600 premature deaths avoided annually. This increase would substantially increase the economic value of health impacts as well, potentially up to $10 billion. Using the Jaffe et al. (2003) effect estimates for asthma ER visits in the population ages 5 to 34 would have increased the estimated number of avoided admissions by around 4.5 times. This suggests that the all-ages estimates based on earlier studies may underestimate impacts in younger populations. Details of the sensitivity analyses examining alternative mortality and morbidity effect estimates are available from the authors.

This analysis has estimated the health benefits of reducing ozone levels in areas with monitored values that exceed the 8-hour ozone standard. The increasing need to understand the public health impacts of air pollution regulations requires the merging of models and data from many disciplines. While necessary, this type of multi-disciplinary methodology is challenging in complexity and scope. Our approach illustrates the integration of models and data and highlights uncertainties inherent in the end results. The result suggests there may be significant health benefits arising from actions that reduce ozone concentrations in non-attainment areas.

The results of our analysis suggest that moving from current monitored ozone levels to full attainment of the 8-hour standard may yield substantial health benefits. We estimate total benefits (including premature mortality) of meeting the standard as reaching up to $5.7 billion (averaged over the three years, 2000-2002). These dollar benefits are associated with average reductions in health effects including more than 800 avoided premature deaths, more than 4,000 avoided hospital admissions, about 500 avoided asthma emergency room visits per year, over one million avoided restricted activity days, and more than 900,000 avoided school absences.

We provide sensitivity analyses to examine key modeling assumptions. In addition, there are other uncertainties that we could not quantify, such as the importance of unquantified effects and uncertainties in the interpolation of ambient air quality. Inherent in any analysis of health impacts are uncertainties in affected populations, health baselines, incomes, effect estimates and other factors. The assumptions used to capture these elements are reasonable based on the available evidence. However, these data limitations prevent a full scale quantitative estimate of the uncertainty associated with estimates of total economic benefits. If one is mindful of these limitations, the magnitude of the benefit estimates presented here can be useful information in expanding the understanding of the public health impacts of attaining the 8-hour ozone standard.

References:

Adams PF, Hendershot GE, Marano MA. 1999. Current Estimates from the National Health Interview Survey, 1996. Vital Health Stat 10(200):1-212.

Agency for Healthcare Research and Quality. 2000. HCUPnet, Healthcare Cost and Utilization Project. Rockville, Maryland. Available: http://www.ahrq.gov/HCUPnet/ [accessed 29 September 2004].

Anderson HR, Atkinson RW, Peacock JL, Marston L, Konstantinou K. 2004. Meta-analysis of time-series studies and panel studies of Particulate Matter (PM) and Ozone (O3): Report of a WHO task group. Copenhagen, Denmark: World Health Organization.

Berger MC, Blomquist GC, Kenkel D, Tolley GS. 1987. Valuing Changes in Health Risks: A Comparison of Alternative Measures. The Southern Economic Journal 53:977-984.

Burnett RT, Smith-Doiron M, Stieb D, Raizenne ME, Brook JR, Dales RE, et al. 2001. Association between ozone and hospitalization for acute respiratory diseases in children less than 2 years of age. Am J Epidemiol 153(5):444-452.

Chen J, Zhao R, Li Z. 2004. Voronoi-based k-order neighbour relations for spatial analysis. ISPRS Journal of Photogrammetry and Remote Sensing 59(1-2):60-72.

Chen L, Jennison BL, Yang W, Omaye ST. 2000. Elementary school absenteeism and air pollution. Inhal Toxicol 12(11):997-1016.

Cody RP, Weisel CP, Birnbaum G, Lioy PJ. 1992. The effect of ozone associated with summertime photochemical smog on the frequency of asthma visits to hospital emergency departments. Environ Res 58(2):184-94.

Dominici F, McDermott A, Daniels M, Zeger SL, Samet JM. 2003. Revised Analyses of the National Morbidity, Mortality, and Air Pollution Study (NMMAPS), Part II: Mortality Among Residents of 90 Cities. In Revised Analyses of Time-Series Studies of Air Pollution and Health. Special Report. Health Effects Institute, Boston, MA, pp. 9-20.

Fairley, D. 2003. Mortality and Air Pollution for Santa Clara County, California, 1989–1996. In Revised Analyses of Time-Series Studies of Air Pollution and Health. Special Report. Health Effects Institute, Boston, MA, pp. 97-106.

Freeman(III), AM. 1993. The Measurement of Environmental and Resource Values: Theory and Methods. Washington, DC: Resources for the Future.

Gilliland FD, Berhane K, Rappaport EB, Thomas DC, Avol E, Gauderman WJ, et al. 2001. The effects of ambient air pollution on school absenteeism due to respiratory illnesses.

Epidemiology 12(1):43-54.

GeoLytics Inc. 2002. Geolytics CensusCD® 2000 Short Form Blocks. CD-ROM Release 1.0. GeoLytics, Inc. East Brunswick, NJ. Available: http://www.geolytics.com/ [accessed 29 September 2004].

Gold C. 1997. Voronoi methods in GIS. In: van Kreveld, M., Nievergelt, J., Roos, T., Widmayer, P. (Eds.), Algorithmic Foundation of Geographic Information Systems. Lecture notes in Computer Science, vol. 1340. Springer Verlag, Berlin, pp. 21– 35.

Hall JV, Brajer V, Lurmann FW. 2003. Economic Valuation of Ozone-related School Absences in the South Coast Air Basin of California. Contemporary Economic Policy 21(4):407-417.

Harrington W, Portney PR. 1987. Valuing the Benefits of Health and Safety Regulation. Journal of Urban Economics 22:101-112.

Industrial Economics Incorporated (IEc). 1993. Memorandum to Jim DeMocker, U.S. Environmental Protection Agency, Office of Air and Radiation, Office of Policy Analysis and Review. September 30. Washington, D.C.

International Classification of Diseases, Ninth Revision. 1979. Hyattsville, MD: National Center for Health Statistics. Available: http://www.cdc.gov/nchs/about/major/dvs/icd9des.htm [accessed 29 September 2004].

Ito K, Thurston GD. 1996. Daily PM10/mortality associations: an investigation of at-risk subpopulations. J Expo Anal Environ Epidemiol 6(1):79-95.

Jaffe DH, Singer ME, Rimm AA. 2003. Air pollution and emergency department visits for asthma among Ohio Medicaid recipients, 1991-1996. Environ Res 91(1):21-28.

Kelsall J, Samet JM, Zeger SL. 1997. Air Pollution, and Mortality in Philadelphia, 1974-1988. Am J Epidemiol 146:750–762.

Kinney PL, Ito K, Thurston GD. 1995. A Sensitivity Analysis of Mortality Pm-10 Associations in Los Angeles. Inhal Toxicol 7(1):59-69.

Levy JI, Carrothers TJ, Tuomisto JT, Hammitt JK, Evans JS. 2001. Assessing the Public Health Benefits of Reduced Ozone Concentrations. Environ Health Perspect 109(12):1215-1226

McDonnell WF, Abbey DE, Nishino N, Lebowitz MD. 1999. Long-term ambient ozone concentration and the incidence of asthma in nonsmoking adults: the AHSMOG study. Environ Res 80(2 Pt 1):110-21.

Moolgavkar SH, Luebeck EG, Hall TA, Anderson EL. 1995a. Air Pollution and Daily Mortality in Philadelphia. Epidemiology 6(5):476-484.

Moolgavkar SH, Luebeck EG, Hall TA, Anderson EL. 1995b. Particulate Air Pollution, Sulfur Dioxide, and Daily Mortality - a Reanalysis of the Steubenville Data. Inhal Toxicol 7(1):35-44.

Moolgavkar SH, Luebeck EG, Anderson EL. 1997. Air pollution and hospital admissions for respiratory causes in Minneapolis St. Paul and Birmingham. Epidemiology 8(4):364-370.

Moolgavkar SH. 2000. Air pollution and daily mortality in three U.S. counties. Environ Health Perspect 108:777-784.

National Research Council (NRC). 2002. Estimating the Public Health Benefits of Proposed Air Pollution Regulations. The National Academies Press: Washington, D.C.

Ostro BD, Rothschild S. 1989. Air Pollution and Acute Respiratory Morbidity - an Observational Study of Multiple Pollutants. Environ Res 50(2):238-247.

Ransom MR, Pope CA. 1992. Elementary School Absences and PM(10) Pollution in Utah Valley. Environ Res 58(2):204-219.

Samet JM, Zeger SL, Kelsall JE, Xu J, Kalkstein LS. 1997. Air Pollution, Weather, and Mortality in Philadelphia 1973-1988. Health Effects Institute. Cambridge, MA.

Schwartz J. 1994a. PM(10) Ozone, and Hospital Admissions For the Elderly in Minneapolis St Paul, Minnesota. Arch Environ Health 49(5):366-374.

Schwartz J. 1994b. Air Pollution and Hospital Admissions For the Elderly in Detroit, Michigan. Am J Respir Crit Care Med 150(3):648-655.

Schwartz J. 1995. Short term fluctuations in air pollution and hospital admissions of the elderly for respiratory disease. Thorax 50(5):531-538.

Smith DH, Malone DC, Lawson KA, Okamoto LJ, Battista C, Saunders WB. 1997. A national estimate of the economic costs of asthma. Am J Respir Crit Care Med. 156(3 Pt 1):787-793.

Stanford R, McLaughlin T, Okamoto LJ. 1999. The cost of asthma in the emergency department and hospital. Am J Respir Crit Care Med 160(1):211-215.

Stieb DM, Burnett RT, Beveridge RC, Brook JR. 1996. Association between ozone and asthma emergency department visits in Saint John, New Brunswick, Canada. Environ Health

Perspect 104(12):1354-1360.

Thurston GD, Ito K. 2001. Epidemiological studies of acute ozone exposures and mortality. *J* Expo Anal Environ Epidemiol 11(4):286-294.

Tolley GS, Babcock L, Berger M, Bilotti A, Blomquist G, Brien M, et al. 1986. Valuation of Reductions in Human Health Symptoms and Risks. Prepared for U.S. Environmental Protection Agency. January. Washington, D.C.

Toulomi G, Katsouyanni K, Zmirou D, Schwartz J. 1997. Short-Term Effects of Ambient Oxidant Exposure on Mortality: A combined Analysis within the APHEA Project. Am J Epidemiol 146:177–183.

U.S. Bureau of the Census. 2001. Statistical Abstract of the United States, 2001, Section 12: Labor Force, Employment, and Earnings, Table No. 521. Washington, DC.

U.S. Department of Education. 1996. The Condition of Education 1996, Indicator 42: Student Absenteeism and Tardiness. National Center for Education Statistics. Washington DC.

U.S. EPA. 1970. Clean Air Act. 40CFR50.

U.S. EPA. 1996a. Review of National Ambient Air Quality Standards for Ozone: Assessment of Scientific and Technical Information. OAQPS Staff Paper. U.S. EPA, Office of Air Quality Planning and Standards. Research Triangle Park, NC. EPA-452\R-96-007. June.

U.S. EPA. 1996b. Air Quality Criteria for Ozone and Related Photochemical Oxidants. Volume III. U.S. EPA, Office of Research and Development. Washington, DC. EPA-/600/P-93/004cF. July.

U.S. Environmental Protection Agency. 1999. The Benefits and Costs of the Clean Air Act, 1990-2010. Prepared for U.S. Congress by U.S. EPA, Office of Air and Radiation/Office of Policy Analysis and Review, Washington, DC, November; EPA report no. EPA-410-R-99-001.

U.S. Environmental Protection Agency, 2003. Latest Findings on National Air Quality: 2002 Status and Trends. Office of Air and Radiation. Washington, D.C., EPA Report no. EPA 454/ K-03-001.

U.S. Environmental Protection Agency, 2004a. Benefits of the Proposed Inter-State Air Quality Rule. EPA 452/-03-001. Prepared by Office of Air and Radiation. Available: http://www.epa.gov/interstateairquality/tsd0175.pdf [accessed 29 September 2004].

U.S. Environmental Protection Agency, 2004b. Final Regulatory Analysis: Control of Emissions from Nonroad Diesel Engines. EPA420-R-04-007. Prepared by Office of Air and Radiation. Available: http://www.epa.gov/nonroad-diesel/2004fr/420r04007.pdf [accessed 29 September 2004].

Vingarzan R. 2004. A review of surface ozone background levels and trends. Atmos Environ 38:3431-3442.

Weisel CP, Cody RP, Lioy PJ. 1995. Relationship between summertime ambient ozone levels and emergency department visits for asthma in central New Jersey. Environ Health Perspect 103(Suppl 2):97-102.

Woods & Poole Economics Inc. 2001. Population by Single Year of Age CD. CD-ROM. Woods & Poole Economics, Inc. Washington, D.C.

World Health Organization (WHO). 2003. Health Aspects of Air Pollution with Particulate Matter, Ozone and Nitrogen Dioxide: Report on a WHO Working Group. World Health Organization. Bonn, Germany. EUR/03/5042688.

Table 1. Distribution of 4th Highest Maximum Daily Average Ozone Values Across Monitors

Range of Ozone Values (ppb)	**Percent of Monitors With Value in Range 2000 (1089 monitors)**	**2001 (1120 monitors)**	**2002 (1146 monitors)**
O3≤84 (in attainment)	64%	61%	44%
84<O3≤90	17%	18%	15%
90<O3≤100	15%	16%	27%
100<O3≤110	3%	4%	11%
O3>110	1%	1%	3%

Table 2. Ozone-Related Health Endpoints Included in Primary and Sensitivity Analyses

Health Effect	Applied Ages	Description	Ozone Metric
Premature Mortality	All	Pooled estimate:	
		Ito and Thurston (1996)	1-hour daily max
		Moolgavkar et al. (1995b)	24-hour daily avg
		Samet et al. (1997)	24-hour daily avg
	All	Sensitivity:	
		WHO (2003)	8-hour average
Respiratory Hospital Admissions	65+	Pooled estimate:	
		Schwartz (1995) - ICD 460-519 (all resp)	24-hour daily avg
		Schwartz (1994a; 1994b) - ICD 480-486 (pneumonia)	
		Moolgavkar et al. (1997) - ICD 480-487 (pneumonia)	
		Schwartz (1994b) - ICD 491-492, 494-496 (COPD)	
		Moolgavkar et al. (1997) – ICD 490-496 (COPD)	
	<2	Burnett et al. (2001)	24-hour daily avg
Asthma-Related ER Visits	All	Pooled estimate:	
		Weisel et al. (1995)	5-hour daily avg
		Cody et al. (1992)	5-hour daily avg
		Stieb et al. (1996)	24-hour daily avg
		Sensitivity:	
	5-34	Jaffe et al. (2003)	8-hour daily max
Other Health Effects			
School Loss Days [a]		Pooled estimate:	
	5-17	Gilliland et al. (2001)	8-hour daily avg
	5-17	Chen et al. (2000)	1-hour daily max
Minor Restricted Activity Days (MRADs)	18-65	Ostro and Rothschild (1989)	24-hour daily avg

[a] Gilliland et al. (2001) studied children aged 9 and 10. Chen et al. (2000) studied children 6 to 11. Based on recent advice from the National Research Council and the EPA Science Advisory Board Health Effects Subcommittee, we have calculated reductions in school absences for all school-aged children based on the biological similarity between children aged 5 to 17.

Table 3. National Average Baseline Incidence Rates

Endpoint	**Source**	**Notes**	**Rate per 100 people per year 4 by Age Group**						
			<18	**18-24**	**25-34**	**35-44**	**45-54**	**55-64**	**65+**
Mortality	CDC Compressed Mortality File, non-accidental accessed through CDC Wonder (1996-1998)	non-accidental	0.025	0.022	0.057	0.150	0.383	1.006	4.937
Respiratory Hospital Admissions.	1999 NHDS public use data files [b]	incidence	0.043	0.084	0.206	0.678	1.926	4.389	11.629
Asthma ER visits	2000 NHAMCS public use data files [c]; 1999 NHDS public use data files [b]	incidence	1.011	1.087	0.751	0.438	0.352	0.425	0.232
Minor Restricted Activity Days (MRADs)	Ostro and Rothschild (1989, p. 243)	incidence	–	780	780	780	780	780	–
School Loss Days	National Center for Education Statistics (1996) And 1996 HIS (Adams et al., 1999, Table 47); estimate of 180 school days per year	all-cause	990.0	–	–	–	–	–	–

[a] The following abbreviations are used to describe the national surveys conducted by the National Center for Health Statistics: HIS refers to the National Health Interview Survey; NHDS - National Hospital Discharge Survey; NHAMCS - National Hospital Ambulatory Medical Care Survey.

[b] See ftp://ftp.cdc.gov/pub/Health_Statistics/NCHS/Datasets/NHDS/

[c] See ftp://ftp.cdc.gov/pub/Health_Statistics/NCHS/Datasets/NHAMCS/

[d] All of the rates reported here are population-weighted incidence rates per 100 people per year. Additional details on the incidence and prevalence rates, as well as the sources for these rates are available upon request.

Table 4. Unit Values for Economic Valuation of Health Endpoints (2000$)

Health Endpoint	Description	Mean Estimate adjusted for Income Growth to 2000 [a b]	Distribution
Mortality	VSL based on 26 studies	$6.5 million per statistical life	The $6.5 million estimate is the mean of a Weibull distribution fitted to the estimates from 26 value-of-life studies identified in the U.S. EPA Section 812 reports (e.g., EPA, 1999) as "applicable to policy analysis." Five of the 26 studies are contingent valuation studies, which directly solicit willingness-to-pay information from surveyed subjects. The remainder are wage-risk studies, which base WTP estimates on estimates of the additional compensation demanded for riskier jobs.
Hospital Admissions	all respiratory, ages 65+	$18,353 per admission	No distributions available. The COI point estimates (lost earnings plus direct medical costs) are based on ICD-9 code level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total COPD category illnesses) reported in Agency for Healthcare Research and Quality, 2000 (www.ahrq.gov).
	all respiratory ages 0-2	$7,741 per admission	
Emergency Room Visits:	Asthma-related	$286 per visit	No distribution available. The COI point estimate is the simple average of two unit COI values (1) $312, from Smith et al. (1997), and (2) $261, from Stanford et al. (1999).
Minor Effects	Minor restricted activity day (MRAD)	$52 per day	Median WTP estimate to avoid one MRAD from Tolley, et al. (1986) . Distribution is assumed to be triangular with a minimum of $22 and a maximum of $83. Range is based on assumption that value should exceed WTP for a single mild symptom (the highest estimate for a single symptom--for eye irritation--is $16.00) and be less than that for a WLD. The triangular distribution acknowledges that the actual value is likely to be closer to the point estimate than either extreme.
School absences		$75 per day	No distribution available.

[a] The derivation of each of the estimates is discussed in the text.
[b] Cost of illness based unit values are not adjusted for income growth as they are based on current costs and wage rates. These include hospital admissions, ER visits, and school absences.

Table 5. Summary of Estimated Annual Health Benefits of Attaining the 8-Hour Standard

Endpoint	2000 Cases	2000 Economic Value (Million 2000$)	2001 Cases	2001 Economic Value (Million 2000$)	2002 Cases	2002 Economic Value (Million 2000$)
Quadratic Rollback						
Premature mortality	560	$3,600	670	$4,400	1,300	$8,400
Hospital admissions – respiratory, adults	1,500	$27	1,900	$34	3,600	$67
Total hospital admissions – respiratory, children	1,700	$13	1,600	$13	2,900	$23
Emergency room visits for asthma	370	$0.11	410	$0.12	750	$0.22
School absences	740,000	$55	780,000	$59	1,400,000	$110
Minor restricted activity days	950,000	$49	1,100,000	$55	2,000,000	$100
Total Economic Value Of Health Changes						
with premature mortality		$3,700		$4,600		$8,700
without premature mortality		$140		$160		$300
Percentage Rollback						
Premature mortality	500	$3,200	590	$3,300	1,160	$7,600
Hospital admissions- respiratory, adults	1,300	$24	1,600	$17	3,200	$60
Total hospital admissions – respiratory, children	1,500	$12	1,500	$3	2,700	$21
Emergency room visits for asthma	330	$0.10	360	$0.05	680	$0.20
School absences	660,000	$50	700,000	$27	1,300,000	$97
Minor restricted activity days	850,000	$44	950,000	$18	1,800,000	$93
Total Economic Value of Health Changes						
with premature mortality		$3,300		$3,400		$7,900
without premature mortality		$130		$70		$270

Table 6. Estimated Average Annual Health Benefits of Attaining 8-Hour Standard (2000 – 2002 Monitor Data)

Endpoint	Age	Cases			Economic Value (Million 2000$)		
		5%	mean	95%	5%	mean	95%
Quadratic Rollback							
Premature mortality	All	290	840	1,600	$500	$5,500	$15,000
Hospital admissions - respiratory, adults	65+	530	2,300	4,600	$10	$43	$84
Total hospital admissions – respiratory, children	0-1	1,100	2,100	3,100	$8.70	$16	$24
Subtotal hospital admissions – respiratory	--	1,600	4,400	7,700	$18	$59	$110
Emergency room visits for asthma	All	180	510	870	$0.05	$0.15	$0.26
School absences	5-17	350,000	970,000	1,700,000	$26	$75	$130
Minor restricted activity days	18-64	670,000	1,400,000	2,000,000	$28	$68	$110
Total Economic Value of Health Changes							
with premature mortality					$570	$5,700	$15,000
without premature mortality					$70	$200	$350
Percentage Rollback							
Premature mortality	All	260	750	1,400	$470	$4,700	$13,000
Hospital admissions - respiratory, adults	65+	470	2,000	4,100	$8.70	$34	$76
Total hospital admissions – respiratory, children	0-1	970	1,900	2,800	$7.70	$12	$22
Subtotal hospital admissions – respiratory	--	1,500	3,900	6,700	$16	$46	$97
Emergency room visits for asthma	All	150	460	770	$0.04	$0.12	$0.23
School loss days	5-17	310,000	890,000	1,500,000	$23	$58	$120
Minor restricted activity days	18-64	610,000	1,200,000	1,800,000	$26	$52	$110
Total Economic Value of Health Changes							
with premature mortality					$530	$4,900	$14,000
without premature mortality					$65	$160	$310

Table 7. Sensitivity of Mean Estimated Annual Health Effects of Attaining the 8-Hour Standard Relative to 2001 Monitor Values, to Ordinality, Attainment Metric Background (AMB), and Hourly Background (HB) (cases)[abc]

Endpoint						
	Ordinality	**1**	**4**	**4**	**4**	**1**
	AMB	**0**	**0**	**40**	**40**	**40**
	HB	**0**	**0**	**40**	**0**	**40**
Quadratic Rollback						
Premature Mortality		1,600	700	300	700	500
Hospital admissions- Respiratory, adults		4,600	2,000	700	1,900	1,300
Hospital admissions – Respiratory, children		3,890	1,730	1,110	1,630	2,010
ER Visits for Asthma		970	430	220	410	400
School Loss Days		1,900,000	840,000	520,000	780,000	950,000
MRAD		2,600,000	1,100,000	430,000	1,100,000	760,000
Percentage Rollback						
Premature Mortality		2,800	1,100	400	600	900
Hospital admissions – Respiratory, adults		8,100	3,100	1,100	1,600	2,500
Hospital admissions – Respiratory, children		6,840	2,620	1,770	1,460	4,010
ER Visits for Asthma		1,900	650	340	360	750
School Loss Days		3,300,000	1,300,000	840,000	700,000	1,900,000
MRAD		4,500,000	1,700,000	660,000	950,000	1,400,000

[a] Sensitivity conducted using the VNA interpolation method with no distance limit.
[b] Ordinality refers to the nth highest value used to determine attainment with the level of the standard. For example, the form of the 8-hour standard specifies the 4th highest maximum 8-hour average. The ordinality in this case is 4. Attainment metric background refers to the assumed level of the attainment standard (4th highest maximum 8-hour average) that would exist in the absence of domestic man-made emissions of ozone precursors. Hourly background refers to the assumed level of ozone at any hour that would exist in the absence of domestic man-made emissions of ozone precursors.
[c] The gray column is the base case analysis presented in Table 5.